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Research Article

Monitoring Burrowing Petrel Populations: A Sampling Scheme for the Management of an Island Keystone Species

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ABSTRACT Burrow-nesting petrels (order *Procellariiformes*) are keystone species in island ecosystems, where they modify habitat through guano deposition and burrow digging. Burrowing petrels are among the most threatened groups of birds, yet robust long-term monitoring data remain scarce because of the financial and logistical constraints of working on offshore breeding islands, the variety of surveying strategies used, and the birds' below-ground breeding behavior. We examined the sampling requirements of monitoring programs to detect changes in the number of breeding pairs of gray-faced petrels (*Pterodroma gouldi*), a common species in northern New Zealand. We first examined the relationship between burrow entrance density and breeding pair density using 4 years of data from 3 large colonies. We then conducted a simulation-based power analysis to assess the ability of different burrow-occupancy sampling regimes to detect changes in breeding bird abundance. Power to detect change was influenced by population growth rates, initial bird density, inter-annual variation in abundance, plot size, number of plots, intervals between surveys, time of year surveys are undertaken, and duration of the monitoring program. Our analyses suggest that, under the most suboptimal monitoring conditions, at least 45 randomly assigned 5-m-radius plots surveyed annually during the incubation period for $\geq$20 years will be required to detect a 1% annual change in breeding bird abundance. Because power will vary depending on project specifications, local conditions, and potential change, we created an online application with over 50,000 combinations of starting parameters (https://landcare.shinyapps.io/petrels). This allows managers to determine the power of different combinations of survey intensities while maintaining consistency and maximizing efficiency. © 2015 The Wildlife Society.

KEY WORDS gray-faced petrels, monitoring, New Zealand, population census, power, *Pterodroma gouldi*, seabirds, simulation analysis.

Burrow-nesting petrels are important in their marine and terrestrial environments as indicators of marine health and engineers of terrestrial ecosystems (Parsons et al. 2008, Smith et al. 2011). An increase in petrel populations is necessary to reinstate island ecosystem functioning after restoration interventions (Jones 2010). Moreover, seabirds are an important source of food and cultural identity for many people around the world (Montevecchi et al. 2007, Lyver et al. 2008). However, burrow-nesting seabirds are among the most threatened group of marine animals (Dulvy et al. 2003, Moller 2009, Croxall et al. 2012). Petrel populations have undergone severe reductions, primarily due to the impacts of introduced mammalian predators to their breeding sites and incidental fisheries by-catch (Burger and Gochfeld 1994, Lewison et al. 2012). Over the past few decades, predator eradication efforts and the implementation of by-catch mitigation measures have increased (Anderson et al. 2011, Keitt et al. 2011). Robust monitoring is, therefore, important for detecting trends in petrel populations, informing adaptive management by measuring the outcome of restoration, and providing estimates for trajectory models under predicted future conditions (Block et al. 2003, Field et al. 2007).

Despite their threat status and importance, reliable long-term population estimates for petrels are scarce because of the technical and logistical challenges associated with implementing an effective monitoring program. Petrels have cryptic nesting behavior, including below-ground nesting and nocturnal colony attendance (Warham 1990). Most species have high inter-annual variability in breeding participation, which makes short-term trend estimation impossible (Newman et al. 2009*a*). Heavily burrowed friable soils make surveying difficult without collapsing burrows (Kennedy and Pachlatko 2012), and accessing island breeding grounds can be expensive and logistically challenging (Schumann et al. 2013). Additionally, there are complex sources of variation associated with measuring seabird

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colonies, which must be incorporated into a sampling design (MacKenzie et al. 2002). These include large inter- and intra-island spatial variation, where colonial petrels nest in clusters within an island and colony size can vary from a few hundred to over half a million individuals (Jones 2000, Rayner et al. 2007, Newman et al. 2009*b*). Furthermore, because petrels nest in burrows, abundance can be estimated using 1) relative indices of abundance (e.g., surface counts before birds enter nests [Renner et al. 2011], nocturnal call counts [Buxton and Jones 2012, Borker et al. 2014], burrow entrance densities [McKechnie et al. 2009]); 2) estimates of breeding attempts via extrapolation of burrow content surveys using either infrared cameras (Lyver et al. 1998) or observation lids over excavated nesting chambers (Rayner et al. 2007); or 3) using mark–recapture methods (Sutherland and Dann 2012). Relative indices of abundance will have wide confidence intervals and require often indefensible assumptions, burrow occupancy estimates are time-consuming and associated with detection error, and mark–recapture may be infeasible for many species (MacKenzie et al. 2002, Mackenzie 2005, Renner et al. 2011).

An effective monitoring program requires a statistically robust design that is able to detect change in the parameter of interest over and above the natural inherent variability of that parameter (i.e., the statistical power; Carignan and Villard 2002, Field et al. 2007). To assess the robustness of a sampling regime, power analyses are often employed to ensure that sample size will be great enough to detect biologically relevant changes but not so great that limited financial resources are wasted (Hatch 2003). The resulting level of statistical power represents the probability of rejecting the null hypothesis when it is false (i.e., detecting a population trend when one actually occurs; Cohen 1988). Without these analyses, a conservation program risks being unable to respond to real population declines (Field et al. 2005).

New Zealand is globally significant in terms of both petrel diversity and introduced predator eradication (Taylor 2000, Towns 2011). However, monitoring programs are rarely implemented for petrels at breeding sites, and when they are, they often suffer from poor design (Buxton 2014). Impetus for designing a consistent national monitoring strategy is evolving as part of national initiatives to monitor status and trends in mainland biodiversity (Lee et al. 2005, Department of Conservation 2010, MacLeod et al. 2012). Both the Department of Conservation and Māori (indigenous peoples of New Zealand) are interested in expanding this monitoring scheme to offshore islands, using petrels as biological indicators (B. Greene and G. Welch, Department of Conservation, unpublished report, Moller 2009). Gray-faced petrels (ōi, tītī, northern muttonbird, *Pterodroma gouldi*) are culturally significant (Lyver et al. 2008) and relatively well studied and widespread in northeastern New Zealand, thus representing an ideal indicator species (Imber 1976, Marchant and Higgins 1990, Lawrence et al. 2014, Whitehead et al. 2014). Our objective was to develop a versatile sampling design for monitoring changes in petrel breeding pair abundance, using gray-faced petrel breeding populations on New Zealand islands as a model.

STUDY AREA

We used burrow entrance counts and occupancy surveys collected as part of a previous study from 3 islands off the northeastern coast of New Zealand's North Island: Moutohorā (148 ha), Ruamāhuanui (21 ha), and Ruamāhuaiti (16 ha, Fig. 1; Whitehead et al. 2014). All study islands have similarly ubiquitous gray-faced petrel distribution, warm-temperate climate, volcanic geology, and post-fire vegetation structure dominated by pohutukawa (*Metrosideros excelsa*; Whitehead et al. 2014). All islands were extensively burned and terraced by Māori and Moutohorā was farmed by Europeans, but all islands are now protected as nature reserves (Sladden and Falla 1928, Edgar 1962, McCallum et al. 1984). The Ruamāhua islands support breeding populations of 5 species of burrow-nesting petrel, including gray-faced petrels, fluttering shearwaters (*Puffinus gavia*), little shearwaters (*P. assimilis*), and common diving petrels (*Pelecanoides urinatrix*), whereas only gray-faced petrels breed on Moutohorā.

METHODS

This research had adhered to animal ethics regulations of the Landcare Research Animal Ethics Committee (Permit nos. 06/02/03 and 10/03/05). To obtain unbiased estimates of burrow occupancy by breeding pairs we assessed burrow occupancy twice a year at each study island. Briefly, 21 (Ruamāhuanui and Ruamāhuaiti) or 27 (Moutohorā) permanent 10-m × 10-m plots were placed randomly over the surface of each island (Whitehead et al. 2014). We used stabilizing wooden boards to prevent burrow collapse when surveying densely burrowed areas. We labeled all burrows where the midline of the entrance fell within plot limits with a cattle tag. A burrow was defined as a cavity greater than 20 cm in length with an entrance greater than 12 cm^2. From 2006 to 2008 (all islands) and, additionally, in 2010 (Ruamāhuanui and Moutohorā) we assessed the occupancy of each burrow during egg incubation in July and the chick rearing period in November. We used a Peeper 2000 flexible video-probe burrow-scope with a head-mounted display to assess occupancy status (Sandpiper Technologies, Manteca, CA, USA). We used burrow-scope detection of an egg or chick to indicate the occupancy of a burrow by a breeding pair. Burrow-scope surveys may miss a portion of burrow occupants, so observed rates of burrow occupancy are likely to be underestimates of true occupancy (e.g., birds overlooked in 34% of sooty shearwater [*Puffinus griseus*] nests in the Snares Islands; Hamilton 2000). Because annual surveys were undertaken during the incubation and chick-rearing periods, we were able to account for the failure to detect a breeding attempt resulting from either burrow-scoping error or nest failure between the egg and chick stages.

For each burrow in each year, we created a 2-period detection history using the detection or non-detection of an egg in period 1 and a chick in period 2. This resulted in 4 possible detection histories: 1,0 indicating that an egg was detected but no chick; 0,1 indicating a chick was detected but no egg; 1,1 indicating both an egg and chick were detected; and 0,0 indicating neither an egg nor chick was detected.

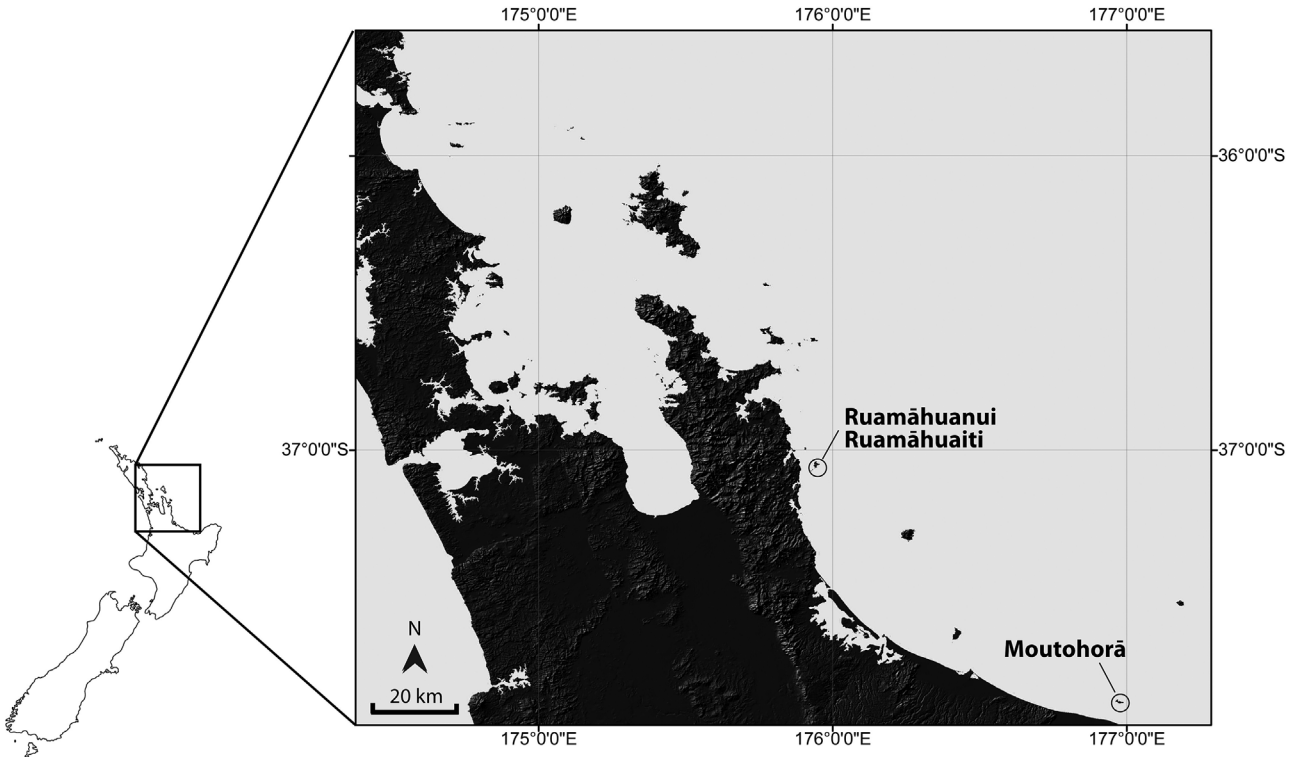

Figure 1. Location of 3 study islands off the northeastern coast of the North Island, New Zealand.

Repeated surveys allowed us to use an occupancy modeling framework to estimate the probability of detecting a breeding pair conditional on the burrow being occupied (MacKenzie et al. 2006).

Occupancy was specified at the burrow level, where the true occupancy, z, of burrow i in year t is given by:

$$Z_{it} = \text{Bern}(\Psi_t)$$

The observed occupancy, y, of burrow i in survey j (egg or chick) in year t is given by

$$y_{ijt} = \text{Bern}(z_{it} \times p_j)$$

where p_j is the probability of detecting a breeding pair. We estimated the number of breeding pairs, B, in each plot k in year t as the sum of the posterior distribution for true occupancy z_{it} of each burrow i and year t, given as:

$$B_{kt} = \sum_{i=1}^{E_{kt}} (z_{it})$$

where the number of burrow entrances, E, in plot k for each year t is:

$$E_{kt} = \text{Poisson}(\mu_t)$$

Separate models were fitted for each of the 3 islands, resulting in island-specific estimates of mean burrow density μ_t and occupancy ψ_t, and, therefore, breeding pair density. We fitted models in JAGS 3.3.0 (Plummer 2003) called from R 3.1.1 (R Development Core Team 2014).

Simulation Power Analysis

We carried out a simulation-based power analysis to assess the ability of different sampling designs to detect a change in the number of breeding pairs under a number of different scenarios. We tested the effects of a range of controllable factors on statistical power: 1) size of sampling plots; 2) number of plots; 3) interval between sampling; and 4) time of year sampling is carried out. We also account for a number of uncontrollable factors that can affect power: 1) the starting density of the population; 2) rate of annual change in the population; 3) natural inter-annual variability in abundance; and 4) burrow occupancy rate by breeding pairs.

We assumed that sampling took place in the incubation or chick-rearing phase of the breeding season. The probabilities of detecting a breeding pair given the burrow was surveyed during incubation or chick-rearing were based on the unbiased estimates from our occupancy modeling (Table 1). We assumed these to be constant for each island and each year. We set a minimum acceptable statistical power threshold of at least 0.8 for detecting a change in breeding bird abundance (hereafter power; Cohen 1988, Freilich et al. 2005). We estimated the effect of the following factors on power in our simulation analyses.

Sampling designs.—We tested the power of the 3 most commonly used plot sizes for petrel surveys in New Zealand: 3-m- and 5-m-radius circular plots, and 10-m × 10-m square plots (Rayner et al. 2007, Newman et al. 2008, Buxton 2014, Whitehead et al. 2014). In New Zealand, the Department of Conservation has indicated that a national monitoring scheme for burrow-nesting petrels on offshore islands would likely involve island visits of 3–7 days, depending on island size, every 5 years (B. S. Greene and G. R. Welch, unpublished report). Based on these constraints, we simulated surveys of 5–100

Table 1. Range of factors simulated to assess power of a monitoring scheme to detect a change in breeding petrel abundance. These include factors that can be manipulated by a monitoring team (survey intensity) and several factors that are out of a manager's control (i.e., annual rate of change in petrel abundance, burrow occupancy, starting breeding bird densities, coefficients of variation of petrel abundance).

Factor		Reference
Survey intensity		
Plot size	3-m radius, 5-m radius, 10-m × 10-m	Rayner et al. (2007), Buxton (2014), Whitehead et al. (2014)
No. plots	5, 10, 15, 20, … 95, 100	B. Greene and G. Welch, unpublished report
No. surveys/decade	2, 5, 10	B. Greene and G. Welch, unpublished report
Survey duration (years)	10, 20	Conservation Act 1987
Time of survey (detection probability)	egg (0.85), chick (0.45)	This study
Annual rate of change in abundance		
Listing as vulnerable	−0.71%	Sæther et al. (2005), Jones et al. (2011*a*)
Listing as endangered	−1.18%	Sæther et al. (2005), Jones et al. (2011*a*)
Listing as critically endangered	−1.89%	Sæther et al. (2005), Jones et al. (2011*a*)
Small rate of change	1.00%	
Passive management	2.00%	Buxton (2014)
Active management	3.00%	Jones et al. (2011*b*)
Burrow occupancy		
Rare to moderately common	0.1	Buxton et al. (2013)
Common	0.2	Buxton (2014)
Abundant	0.4	Whitehead et al. (2014)
Breeding pair density		
Low, mid, high	0.05, 0.1, 0.15, 0.2	
Coefficient of variation (CV)		
Low annual variation	10	
High annual variation	25	

sampling plots, in increments of 5, and survey frequencies of once, twice, and 5 times per 10-year sampling program (Table 1). We used 10- and 20-year sampling periods because a decade is longer than the mean age at first breeding of all burrow-nesting petrel species (Marchant and Higgins 1990, Warham 1990, Brooke 2004) but falls within New Zealand's conservation management strategy timeframe (Department of Conservation 2013). Finally, we simulated single surveys in the egg or chick period of the breeding season.

Trend scenarios.—We tested the power of each simulated design to detect 6 potential population growth trajectories (i.e., effect sizes [Cohen 1988]; Table 1):

1. A decline of 30% over 3 generations, representing the International Union for the Conservation of Nature (IUCN) criterion for listing a species as vulnerable. We estimated gray-faced petrel generation time, T using:

$$T = \propto + \left(\frac{S}{1-S}\right)$$

   where $\propto$ is the average age at first reproduction and S is the adult apparent survival rate (Sæther et al. 2005). Using $S = 0.89$ and $\propto = 6$ (Marchant and Higgins 1990; Jones et al. 2011*a*, 2015*b*) we estimated T to be 14.1 years (range: 11.5–16.7). A 30% decline over 3 generations was, therefore, equivalent to a 0.71% mean annual decline.
2. A mean annual decline of 1.18%, representing the IUCN criterion for listing a species as endangered (equivalent to a decline of 50% over 3 generations).
3. A mean annual decline of 1.81%, representing the IUCN criterion for listing a species as critically endangered (equivalent to a decline of 80% over 3 generations).
4. A 1% annual increase in breeding pair abundance, representing a low level of population growth and based on recent estimates of the annual growth rate of the Moutohorā population (Jones et al. 2015*a*).
5. A 2% annual growth rate representing passive recovery of petrel populations after restoration interventions, based on differences in burrow density along a chronosequence of islands after Pacific rat (*Rattus exulans*) eradication (Buxton 2014).
6. A 3% annual growth rate representing actively managed recovery after restoration interventions (Jones et al. 2011*b*). Active management involves enhancing or reintroducing populations using techniques such as social attraction and translocation.

Burrow occupancy estimates.—We used 3 representative burrow occupancy rates of 0.1, 0.2, and 0.4 to represent low, medium, and high occupancy of burrows, respectively, by breeding pairs (Table 1).

Breeding pair densities and inter-annual variation.—We simulated breeding pair abundance as equivalent densities according to different sized sampling plots. We used densities 0.05, 0.1, 0.15, and 0.2 breeding pairs/m^2. However, petrels are intermittent breeders, where the number of individuals attending a colony and participating in breeding can vary between years, depending on external factors (Cubaynes et al. 2011). Thus, in our simulations, we used 2 levels of variation in annual breeding pair density around the mean trend line to represent a low (CV = 10%) and moderate (CV = 25%) level of natural variation. We used a constant coefficient of variation because seabird count data generally have high correlations between mean and standard deviation (Hatch 2003).

For each set of parameter values, the mean annual density of breeding pairs was simulated for each year $t = 1,\ldots,10$ as $d_t = d_0 \times \lambda^t$, where d_0 was the specified starting density and λ^t was the specified annual rate of change in breeding bird abundance. To include natural inter-annual variation in

petrel density, simulations for each year included a random variate from a normal distribution $D_t = \text{Norm}(d_t, \text{CV} \times d_t)$, where the coefficient of variation was specified as an input parameter. We calculated burrow density as the density of breeding pairs D_t divided by the specified occupancy ψ. We carried out simulations in R 3.1.1 (source code available online in Supporting Information at www.wildlifejournals.org). We simulated survey data using a 5-stage process.

1. We delineated a standardized burrowed area of 18 ha (note: altering the size of this arbitrarily chosen area size did not change results).
2. We distributed burrows across the area using a spatial clustering method. First we divided the standardized area into 25 m × 25-m grids. We chose 25 m because the resulting burrow distribution reflected the distribution of burrows in our data. We assigned each grid a random probability of containing a burrow, and subsequently, assigned burrows to a grid based on grid probabilities.
3. We randomly positioned each burrow within the grid.
4. The true occupancy state of each burrow z was the result of a random draw with probability ψ (see Supporting Information online). The observed occupancy for each burrow was the result of a random draw with probability zP where z is the true occupancy state and P is the aforementioned probability of detecting a breeding pair by proxy.
5. For years where sampling was scheduled to occur, we located the specified number and size of sampling plots randomly across the standardized area. We fitted the simulated survey data with a generalized linear model with Poisson errors:

$$C_i \sim \text{Poisson}(\lambda_i)$$

$$\text{Log}(\lambda_i) = a + bX_i$$

where C_i is the count of either burrows or occupied burrows per plot (depending on the analysis being carried out), X_i is the year, a is the intercept, and b is the coefficient for the annual change.

We simulated each combination of parameters 1,000 times, where the resulting power was the proportion of simulations where the annual change b was statistically significant ($\alpha = 0.05$). Because we had over 50,000 combinations of starting parameters and sampling intensities, we present only general trends and examples as results. The full simulation results are available as user-definable graphics via the online R application Shiny (Chang et al. 2015). This allows users to define their own survey parameters and view the simulated effects on survey power (available at https://landcare.shinyapps.io/petrels).

RESULTS

Survey Data

Burrow density ranged across years and islands, from 0.12 burrows/m^2 (Moutohorā in 2006) to 0.32 burrows/m^2 (Ruamāhuanui in 2010; Fig. 2). The mean probability of breeding pair occupancy/burrow ranged from 0.73 (Moutohorā in 2006) to 0.36 (Ruamāhuaiti 2008; Fig. 2). The mean probability of detecting a breeding pair of birds given they were present in a burrow was 0.85 for the surveys carried out during incubation and 0.45 for surveys carried out during chick rearing. Corresponding mean breeding pair densities were similar within and among islands, with a mean of 0.11 breeding pairs/m^2 on Moutohorā, 0.09 breeding pairs/m^2 on Ruamāhuaiti, and 0.12 breeding pairs/m^2 on Ruamāhuanui. On Moutohorā and Ruamāhuanui, mean burrow density increased over time and mean burrow occupancy declined; in combining these to estimate breeding pair densities no clear trend emerged (Fig. 2). These results suggest a weak relationship between burrow entrance density and breeding pair density.

Simulation Power Analysis

Power to detect a change in breeding bird abundance was greater with an increasing plot area across all simulation analysis parameters (Fig. 3). Increasing circular plot size from 3-m to 5-m radius led to a mean increase in power of 0.14 (95% credible interval 0.05–0.25). However, the mean increase in

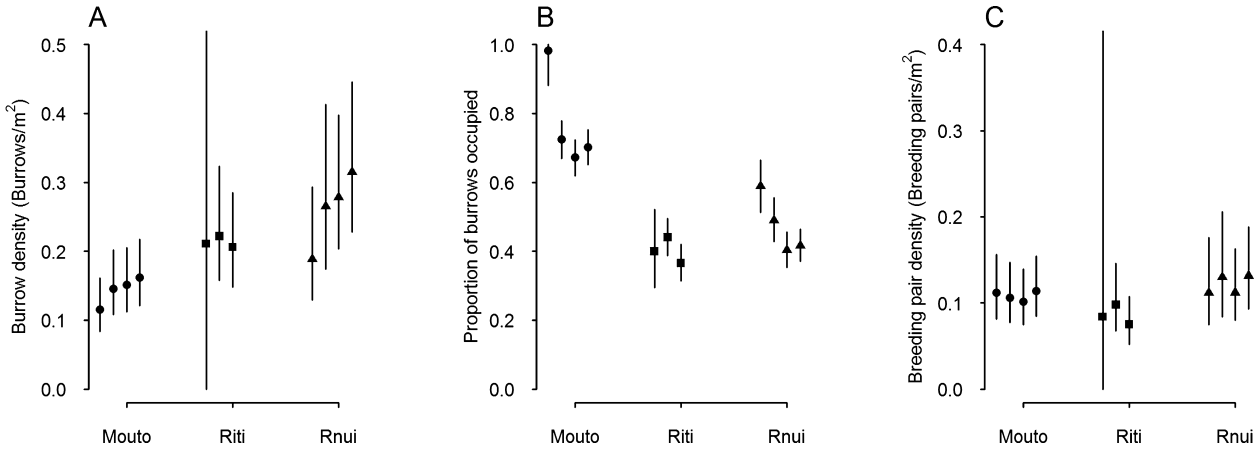

Figure 2. Mean burrow density (A), true occupancy rates (estimated by measuring occupancy during the incubation and chick-rearing periods (B), and density of breeding gray-faced petrels (C) over 3 (2006–2008; Riti) or 4 years (2006–2008, 2010) on 3 islands off the northeastern coast of New Zealand's North Island: Moutohorā (Mouto), Ruamāhuaiti (Riti), and Ruamāhuanui (Rnui). Bars represent 95% credible intervals.

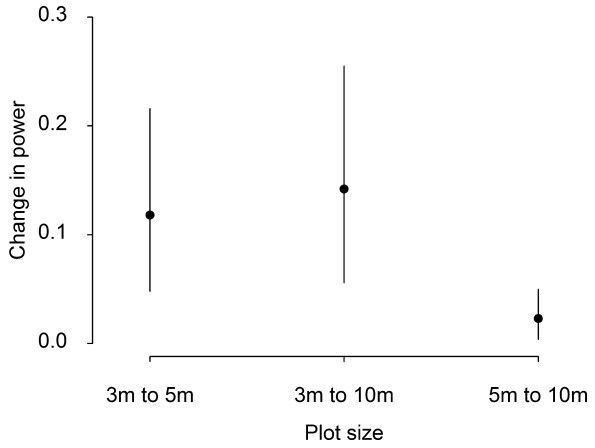

Figure 3. Mean change in power (±95% CI) when plot size increased from 3-m radius (3 m) to 5-m radius (5 m) circular plots and to 10-m × 10-m square plots (10 m) in simulated surveys of gray-faced petrels on islands off the northeastern coast of New Zealand. We averaged change in power across the entire range of sampling effort, breeding pair density, burrow occupancies, annual changes in abundance, and variation around breeding pair abundance.

power from 5-m radius to 10-m × 10-m plots was only 0.02. (0.01–0.05). Because of the large increase in time and logistics required to sample a 10-m × 10-m plot compared with the smaller plots, for a relatively small increase in power, compared to the relatively large power gain from 3-m-radius to 5-m-radius plots, we henceforth focus on the results of monitoring regimes using 5-m-radius plots only.

Across all combinations of sampling parameters, starting burrow occupancy had no discernible effect on power (Fig. 4). We, therefore, report the results from simulations with starting occupancy fixed at 0.40.

Power decreased with 1) increasing time between surveys; 2) fewer plots; 3) when sampling was carried out in the chick-rearing period; 4) decreasing breeding pair density; and 5) increasing inter-annual variation in breeding pairs. On average sampling during the chick-rearing period resulted in a power that was 0.065 (95% CI = 0.01–0.13) lower than sampling in the incubation period. Of the 720 combinations of sampling intensities and starting parameters surveyed over 10 years (Figs. 5 and 6), we found only 48 able to detect changes in abundance with a power of at least 0.8 when sampling fewer than 100 plots. The majority of these required annual sampling of at least 80 plots during the incubation period. Small mean changes in breeding pair numbers were hard to detect without a large effort; we were unable to generate a monitoring scheme with fewer than 100 plots that had a probability of at least 0.8 to detect population growth rates between –1.2% and 1.0 within a 10-year sampling program. When we increased the sampling program length to 20 years, power increased markedly and almost all monitoring scenarios were able to detect our simulated levels of change with power of at least 0.8 (Figs. 7 and 8). However, we were still unable to generate a monitoring scheme with fewer than 100 plots that could detect a small annual population decline of 0.7% (the IUCN criterion for vulnerable) with power of 0.8.

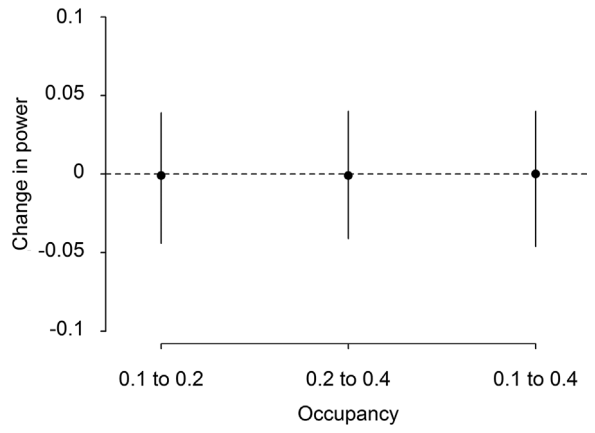

Figure 4. Mean change in power (±95% CI) when increasing initial burrow occupancy rate from 0.1 to 0.2 to 0.4 in simulated surveys of gray-faced petrels on islands off the northeastern coast of New Zealand. We averaged change in power across the entire range of sampling effort, breeding pair density, plot sizes, annual changes in abundance, and variation around breeding pair abundance.

DISCUSSION

Robust ecological monitoring is essential for detecting changes in wildlife abundance. Changes in counts, indices, or estimates of populations serve as guides for documenting the effects of management activities or signaling incipient population decline (Gibbs et al. 1998). Despite the central role that burrow-nesting seabirds play in marine and terrestrial ecosystems and the large proportion of these species that are vulnerable to extinction, consistent long-term monitoring strategies have yet to be devised. Simulation analysis assessing the ability of burrow-occupancy sampling regimes to detect changes in breeding bird abundance revealed that power to detect change was influenced by a range of controllable and uncontrollable factors.

Petrel population estimates are a mix of anecdote, observational best guesses, or inventory surveys, rather than replicated monitoring based on a priori analyses of the statistical requirements to detect population changes (Taylor 2000, Greene 2012, Buxton 2014). Where colonies are small, each burrow may be monitored (e.g., black petrel [*Procellaria parkinsoni*] and Westland petrel [*Procellaria westlandica*]; Bell et al. 2013, Wood and Otley 2013). If these small populations grow, such intensive monitoring will no longer be realistic and sub-samples must be taken (Witmer 2005). Some studies have used search transects traversing a colony area (e.g., flesh-footed shearwater [*Puffinus carneipes*]; Baker et al. 2010, Waugh and Taylor 2012). Although transects are feasible for smaller areas, they are not practical in large or particularly dense colonies, where monitoring would be prohibitively time-consuming and chances of burrow collapse increase (Bancroft 2009). Seabird monitoring around New Zealand is a poorly coordinated and inconsistent activity, with studies carried out by a number of different research institutes, government agencies, non-governmental organizations, and universities (Marsh and Trenham 2008, Buxton 2014). Because most studies use different monitoring

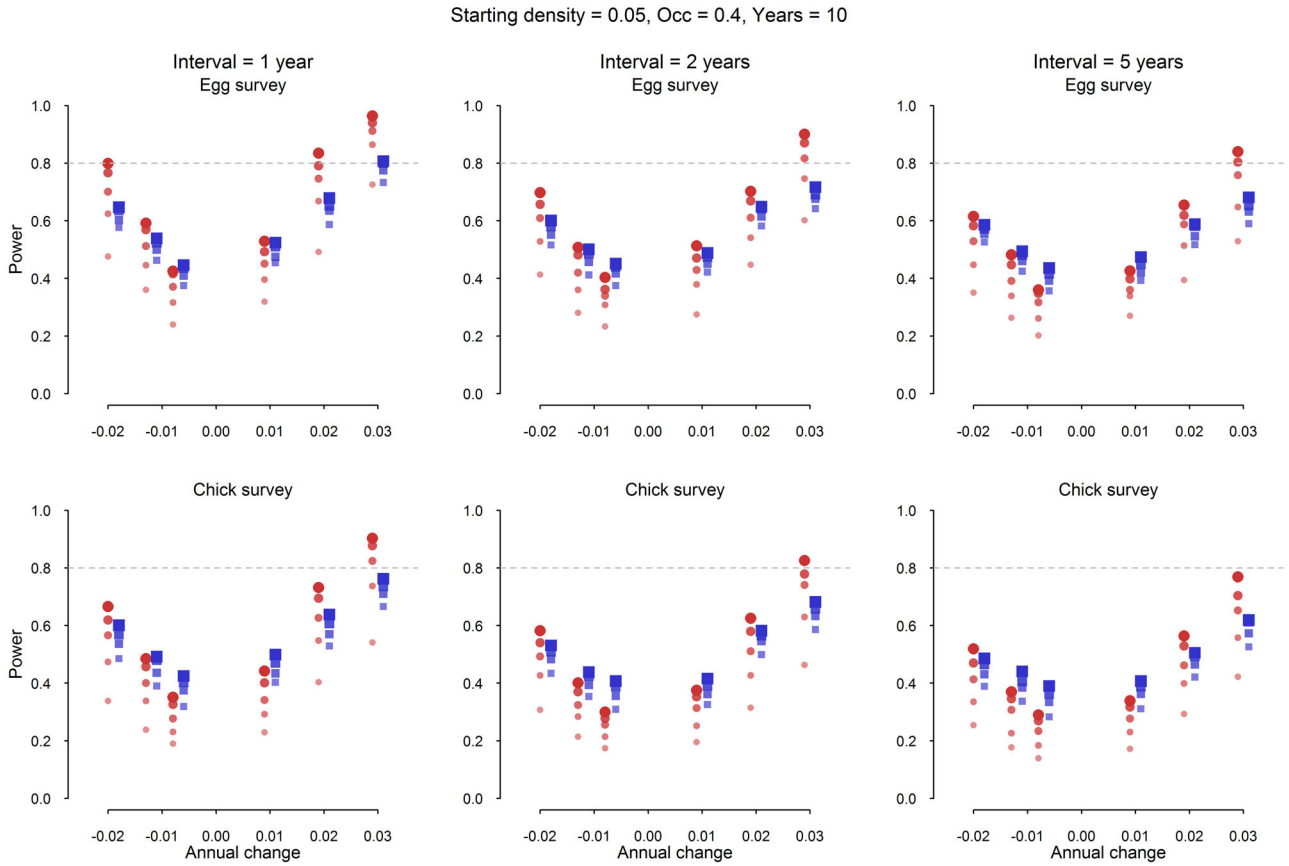

Figure 5. Power of 10-year surveys of 5-m-radius plots with a starting density of 0.05 breeding pairs/m^2 and a burrow occupancy rate (Occ) of 0.4 over a range of annual changes in abundance based on simulated surveys of gray-faced petrels on islands off the northeastern coast of New Zealand. Power of surveys performed in the incubation period is shown in the top 3 graphs, and those in the chick-rearing period are on the bottom 3 graphs. Points increasing in size represent increasing numbers of plots from 20 to 100. Red circles represent CV = 0.1, and blue squares are CV = 0.25.

methodologies, data are not comparable, highlighting the importance of developing a consistent and robust sampling strategy that can be used by any group.

Petrel burrows provide temporally and spatially stable indices of abundance relative to the duration of most surveys (Rayner et al. 2007, McKechnie et al. 2009). Perhaps the most commonly used technique to estimate breeding population size has been to assume that each burrow represents a breeding pair or to assume a constant occupancy rate, estimate burrow density in representative samples, and extrapolate these estimates to the whole colony area (Priddel et al. 2006, Scott et al. 2009). Inferring real relative changes in breeding bird abundance based on a change in burrow entrance abundance assumes that there is a constant, linear, or monotonic increasing relationship between the 2 metrics (i.e., burrow density increases when breeding bird density increases; Caughley and Sinclair 1994). The results of our occupancy modeling did not indicate a consistent relationship between burrow density and breeding pair density (Fig. 2), suggesting that this may not be a reliable indicator of breeding population change over the timespan of our surveys.

Developing a Survey Strategy

Survey accuracy increases if burrow occupancy is examined, as opposed to relying on indices of abundance. However, examining the contents of large numbers of burrows is far more time consuming and expensive. Finding an optimal solution that balances the budget and scale of a petrel monitoring project with statistical power is essential for long-term efficacy (Caughlan and Oakley 2001). We considered a number of survey characteristics that can be modified to increase statistical power.

Number and size of plots.—Generally, we found that power to detect a change in petrel breeding pairs increased with increasing numbers of plots (Figs. 5–8). In many cases (e.g., most survey designs with 3-m-radius plots) >100 plots were required to attain power of 0.8, and in all cases ≥30 plots were required. Generally, surveying >100 plots of 5-m radius would be impractical, requiring high levels of disruption within a colony. Preliminary planning of a national petrel monitoring program in New Zealand indicates that surveys will occur over a maximum of 7 days, depending on island size and colony extent (B. S. Greene and G. R. Welch, unpublished report). Previous surveys suggest that a maximum of 100 5-m-radius circular plots could be measured by a team of 2 within a week, with >1 burrowscope required to measure anything over 30, 10-m × 10-m plots (Buxton 2014).

We tested commonly used plot sizes of ≤100 m^2. We imposed this upper limit because the elevated risk of burrow

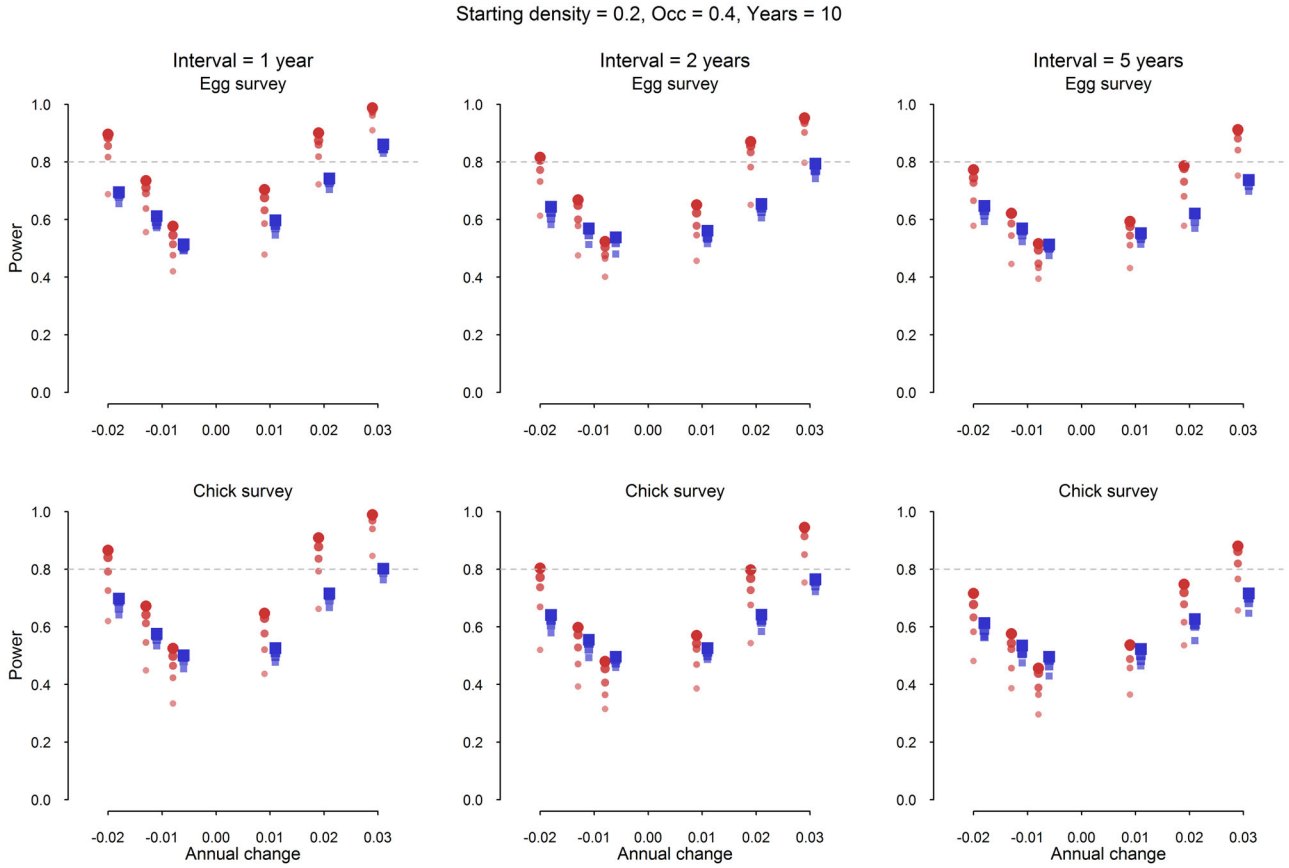

Figure 6. Power of 10-year surveys of 5-m-radius plots with a starting density of 0.2 breeding pairs/m^2 and a burrow occupancy rate (Occ) of 0.4 over a range of annual changes in abundance based on simulated surveys of gray-faced petrels on islands off the northeastern coast of New Zealand. Power of surveys performed in the incubation period is shown in the top 3 graphs, and that in the chick-rearing period is in the bottom 3 graphs. Points increasing in size represent increasing numbers of plots from 20 to 100. Red circles represent CV = 0.1, and blue squares are CV = 0.25.

collapse associated with larger plots and transects in dense colonies compared with the ease of placing stabilizing wooden boards on paths to and around smaller marked plots makes the use of large plots impractical for repeated visits (West and Nilsson 1994). We found that, when averaged among all survey parameters, power increased with increasing plot size (Fig. 3). However, the increase in power from a 5-m-radius plot to a 10-m × 10-m plot was only 0.02, with confidence intervals almost overlapping 0. Considering the small increase in power but large relative increase in survey effort, logistics and costs required for a 100-m^2 square plot compared with a 5-m-radius plot, we recommend using the latter in seabird colony surveys.

Survey duration, frequency, and season.—We constrained survey programs to a 10- or 20-year duration because decadal time periods fit within typical species management plans in New Zealand (Department of Conservation 2013). Although power increased with a longer survey program, we were unable to generate a sampling scheme with power of 0.8 to detect the small (0.7%) annual changes in breeding bird numbers by monitoring fewer than 100 plots (Figs. 5–8). Burrow-nesting petrels generally exhibit k-selected life-history strategies, with long life spans and generation times, and late ages at first reproduction (Warham 1996, Schreiber and Burger 2001). Slow life-history characteristics may result in the need for longer survey duration to detect the annual population changes that, although small, could lead to a species being classified as vulnerable under IUCN criteria if the trend persists.

We found that power increased with shorter intervals between surveys, with the highest power to detect change in petrel abundance when surveys were carried out annually. Program duration also influenced power. If surveys were carried out every 5 years, in a 10-year program using <100 plots during the incubation period, only large annual increases in abundance (≥3%) could be detected with power of 0.8. After 20 years of the same program, smaller annual increases (≥1%) and annual declines of (≥1.9%) in abundance could be detected with power of 0.8.

We also found that surveys during the incubation period had greater power to detect change than those in the chick-rearing period. This is because of the greater probability of detecting a breeding pair using the presence of an egg in a burrow than that of a chick. For burrow-nesting petrels, chicks tend to be easier to detect than eggs (Newman et al. 2009*a*, Southwell et al. 2011, Whitehead et al. 2014). However, if the presence of a chick is used as an indicator of breeding pair abundance, mortality during the egg stage will have had an influence on the detection probability of a breeding attempt. High mortality during the egg-early chick stage, such as that resulting from the

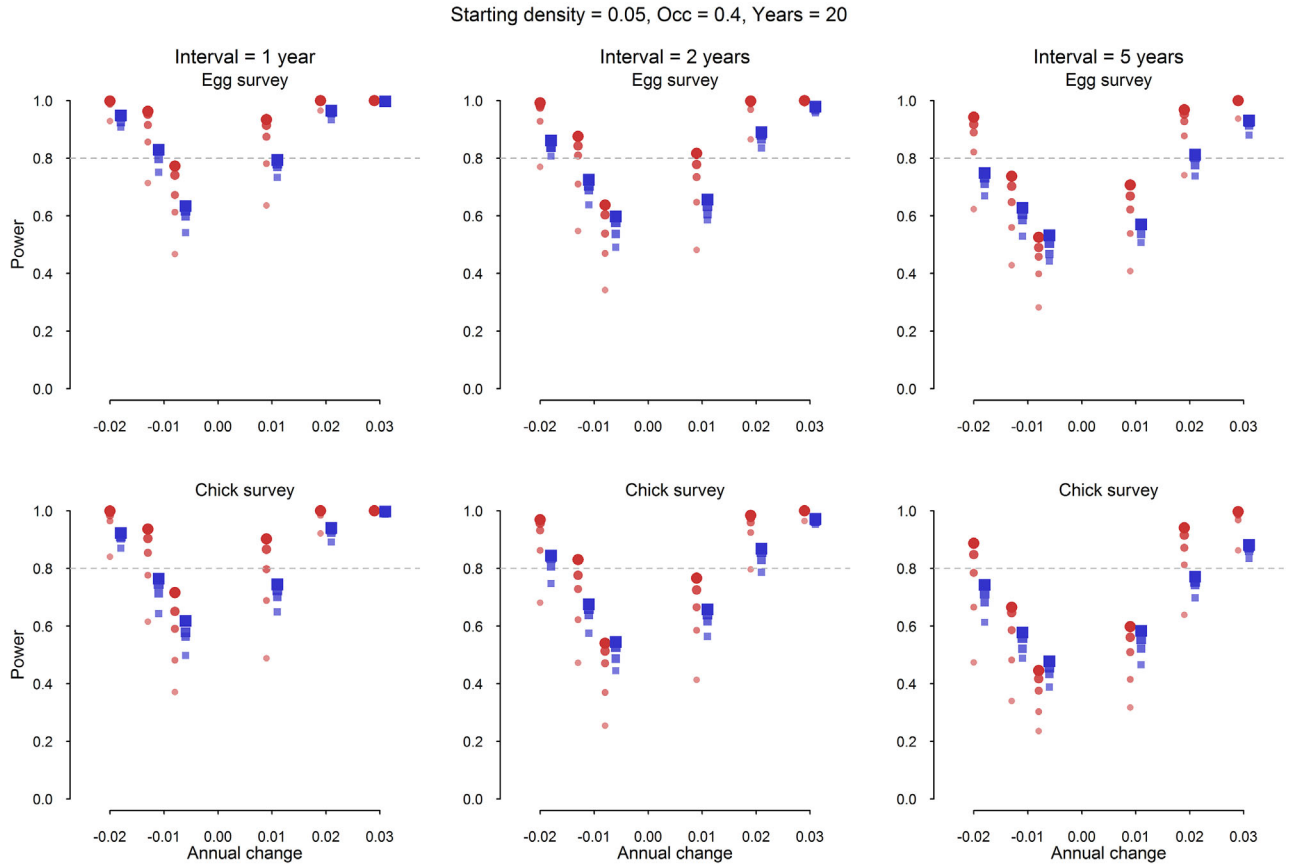

Figure 7. Power of 20-year surveys of 5-m-radius plots with a starting density of 0.05 breeding pairs/m^2 and a burrow occupancy rate (Occ) of 0.4 over a range of annual changes in abundance based on simulated surveys of gray-faced petrels on islands off the northeastern coast of New Zealand. Power of surveys performed in the incubation period is shown in the top 3 graphs, and that in the chick-rearing period is in the bottom 3 graphs. Points increasing in size represent increasing numbers of plots from 20 to 100. Red circles represent CV = 0.1, and blue squares are CV = 0.25.

impacts of introduced predators (Imber 1976), would decrease detectability of a breeding pair using data from chick surveys only. Furthermore, some species or habitats may be associated with higher detection error due to the increased difficulty of finding burrow occupants using a burrow-scope. We, therefore, recommend preliminary surveys during both the egg and chick stages to estimate detection error at a monitoring site, or performing surveys during the egg stage using 0.85 as a best-case-scenario estimate of detection error to determine power.

Although determining survey effort is influenced by a balance between cost-effectiveness and power, it is ultimately defined by a monitoring project's objective (Legg and Nagy 2006, Field et al. 2007). For petrels, monitoring objectives include determining whether a species that is suspected to be declining meets IUCN Red List criteria, tracking the recovery of a population after the removal of a threat (e.g., eradicating non-native predators from breeding grounds), using changes in abundance as an indicator of marine health, or determining if harvest levels are sustainable (Taylor 2000, Parsons et al. 2008, Buxton 2014, Jones et al. 2015*a*). Each monitoring objective may require specific survey criteria. For example, if petrel abundance is to be used as an indicator of marine conditions, surveys should occur annually, in which case other survey characteristics could be altered to maximize power. Although a survey's design features can be modified depending on its objectives and constraints, we identified additional attributes that affected the power of a survey, all of which are likely to be unknown at the outset of monitoring and may change during the program's lifetime. These include population growth rates, initial breeding pair densities, and the degree of inter-annual variation in abundance. In designing a program, it would therefore be wise for managers to assume worst-case conditions (e.g., low breeding bird density, high inter-annual variation) to ensure detection of trends but to allow for adaptive modification of the survey should early data suggest that less effort would be sufficient to meet the program's goals. An awareness of potential changes in parameters and how they may affect power at the outset of a monitoring program allows objectives to be altered without breaching the integrity of a monitoring plan (Ringold et al. 1996). Because of the complex and dynamic suite of variables that must be considered when planning a monitoring program, we integrated our simulations into an online application to display power analysis results. This is intended to allow managers to input combinations of survey effort and starting parameters and determine the resulting power of each potential design.

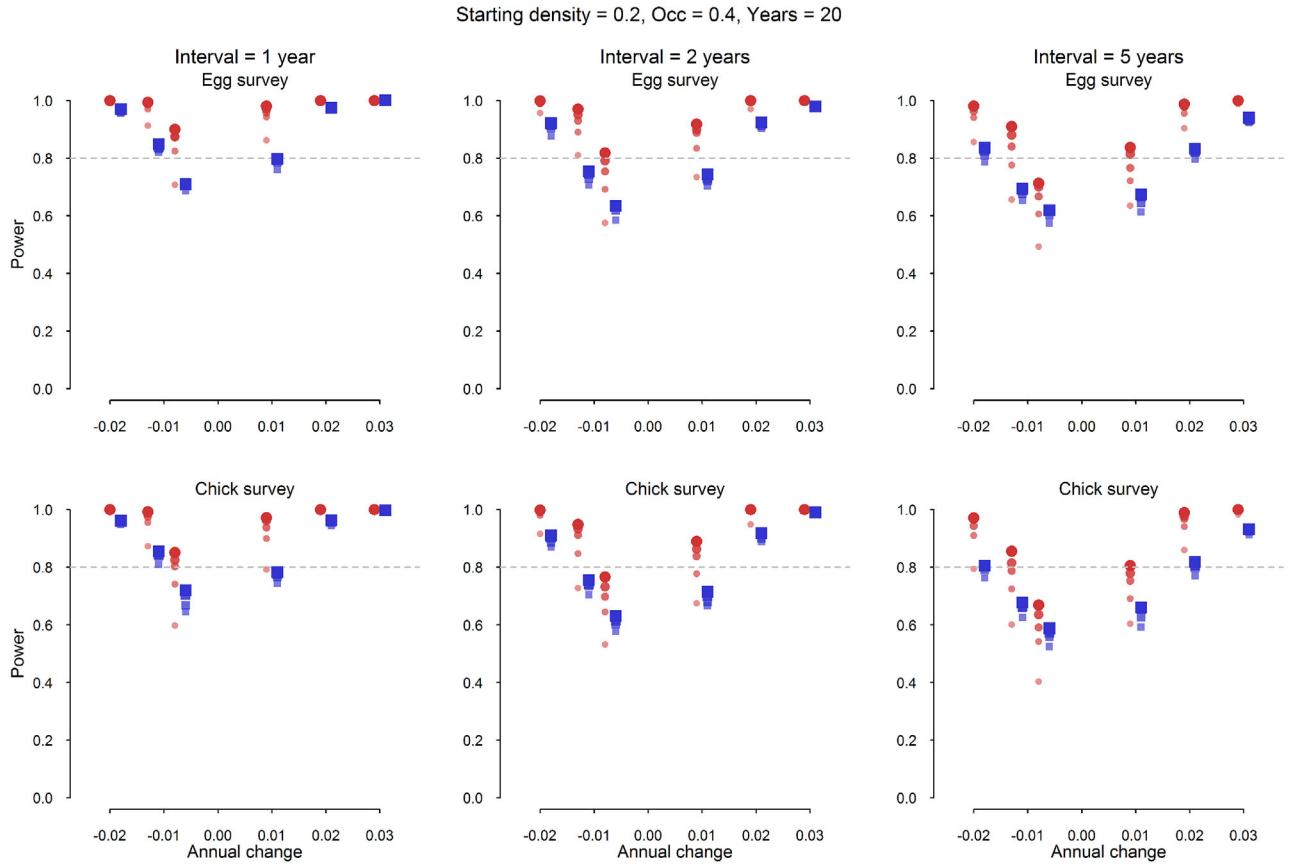

Figure 8. Power of 20-year surveys of 5-m-radius plots with a starting density of 0.20 breeding pairs/m^2 and a burrow occupancy rate (Occ) of 0.4 over a range of annual changes in abundance based on simulated surveys of gray-faced petrels on islands off the northeastern coast of New Zealand. Power of surveys performed in the incubation period is shown in the top 3 graphs, and that in the chick-rearing period is in the bottom 3 graphs. Points increasing in size represent increasing numbers of plots from 20 to 100. Red circles represent CV = 0.1, and blue squares are CV = 0.25.

Other Monitoring Program Considerations

Logistical challenges and complex sources of variation involved in surveys of burrow-nesting seabirds may preclude the use of conventional monitoring schemes. In the early stages of a monitoring program, we recommend setting aside a proportion of plots as reference sites, to be monitored more frequently (Sutherland and Dann 2012). In this way variation in burrow occupancy, detection error, initial breeding bird density, and a potential relationship between burrow entrance density and breeding pair abundance can be estimated. Results from these rigorous subsets can then feed back into survey planning in an adaptive monitoring framework (Lindenmayer and Likens 2009), where the monitoring objective and survey intensity are adjusted accordingly.

Our simulation power analyses considered only spatially defined colonies or burrowed areas rather than entire island surfaces. However, evidence suggests that growing petrel colonies increase in density and spatial extent (Kildaw et al. 2005, Buxton et al. 2016). Thus, if a program's objective is to monitor the recovery of an island population, managers may need to consider monitoring burrowed and non-burrowed areas. Monitoring at the island level introduces complex levels of variance, including spatial clustering, island size, and interaction with habitat features that are beyond the scope of this study but represent important new avenues for research.

We acknowledge that survey design is only 1 step in devising a successful monitoring program. Well-defined and tractable monitoring objectives based on sound conceptual models paired with long-term funding are key features of an effective monitoring strategy (Caughlan and Oakley 2001, Field et al. 2005). Furthermore, a well-developed partnership among seabird scientists, indigenous resource guardians, agency resource managers, and community groups will be key to the stability of any national monitoring strategy (Moller et al. 2004, Lindenmayer and Likens 2010). Finally, as technology continues to develop, lower-cost indices of seabird abundances may arise and could eventually be incorporated into monitoring strategies. Already, new tools and concepts are being tested and applied, such as automated acoustic sensors (Buxton and Jones 2012), radar (Gauthreaux and Belser 2003, Zaugg et al. 2008), and complex modeling (Tavecchia et al. 2009, Gimenez et al. 2012), offering great promise for future monitoring of burrow-nesting seabirds.

MANAGEMENT IMPLICATIONS

Our analyses suggest that using burrow entrance density as a relative index of abundance may be insufficient to distinguish

most trends in breeding bird abundance. For surveys of burrow occupancy using infrared cameras, a study design able to detect a 1% change in breeding pair numbers under worst-case-scenario starting parameters included measuring at least 45 plots of 5-m radius surveyed annually during the incubation period for ≥20 years. Because factors that affect power may be unknown at the start of a monitoring program and beyond the control of a monitoring team, the use of an online application (e.g., https://landcare.shinyapps.io/petrels) can guide managers as to the resulting power of different combinations of survey effort without compromising the consistency of the program. Although the simulations are based on data on gray-faced petrels, we suggest that this application will provide a useful starting point for those responsible for designing surveys of burrowing animals, with little pre-existing information on the relationships between survey components and likely effectiveness.

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